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# Ecological Correlates of Community Structure in Seagrass-Associated Fishes in North Biscayne Bay and Port of Miami, Florida

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Thesis of  
Elizabeth F. Colhoun

Submitted in Partial Fulfillment of the Requirements for the Degree of

Master of Science

M.S. Marine Biology

Nova Southeastern University  
Halmos College of Natural Sciences and Oceanography

May 2018

Approved:  
Thesis Committee

Major Professor: Paul Arena

Committee Member: Christopher Blonar

Committee Member: Andre Daniels

HALMOS COLLEGE OF NATURAL SCIENCES AND  
OCEANOGRAPHY

Ecological Correlates of Community Structure in Seagrass-Associated  
Fishes in North Biscayne Bay and Port of Miami, Florida

By:  
Elizabeth Colhoun

Submitted to the Faculty of Nova Southeastern University Oceanographic  
Center in partial fulfillment of the requirements for the degree of Master of  
Science with a specialty in:  
Marine Biology

Nova Southeastern University

Winter 2018

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Andre M. Daniels

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## ABSTRACT

Seagrass habitats are critical habitat for many fish species and are currently threatened by anthropogenic and natural factors, such as coastal development, pollution, global climate change, and sea level rise. There are few studies that have tracked long-term changes in seagrass habitat and their associated fish communities. This project addressed this need using data collected by the United States Geological Survey (USGS) from two South Florida sites, North Biscayne Bay, FL (NBB) and Port of Miami, FL (POM). The USGS sampling was part of ongoing monitoring projects designed to assist future management decisions that would enhance the protection of these valuable habitats. Data were collected biannually at the conclusion of the dry (April) and wet (September) seasons from 30 cells at each site. In each cell, the data collected included: six replicates for seagrass species and cover, five sweep net collections for fish species and abundance, as well as abiotic variables (water temperature, salinity, turbidity, water depth, and sediment depth). A distinct loss in fish and seagrass species were observed, particularly between the years of 2011-2014. These years coincided with several events including: the Port Miami Deep Dredge (PMDD) project during the years 2013-2015; periods of drought; and major storm events. Changes in fish community structure over this time period were largely driven by loss of species and increased homogenization of fish communities at both locations. More specifically, the NBB community shifted to resemble that of POM by 2014. These changes mirrored the loss of seagrass cover at both locations. Further studies are required to assess the extent to which ongoing dredging activities and other factors might be affecting seagrass cover, which ultimately affect fish communities.

**KEYWORDS:** seagrass coverage, fish communities, North Biscayne Bay, FL, Port of Miami, FL, multivariate statistics

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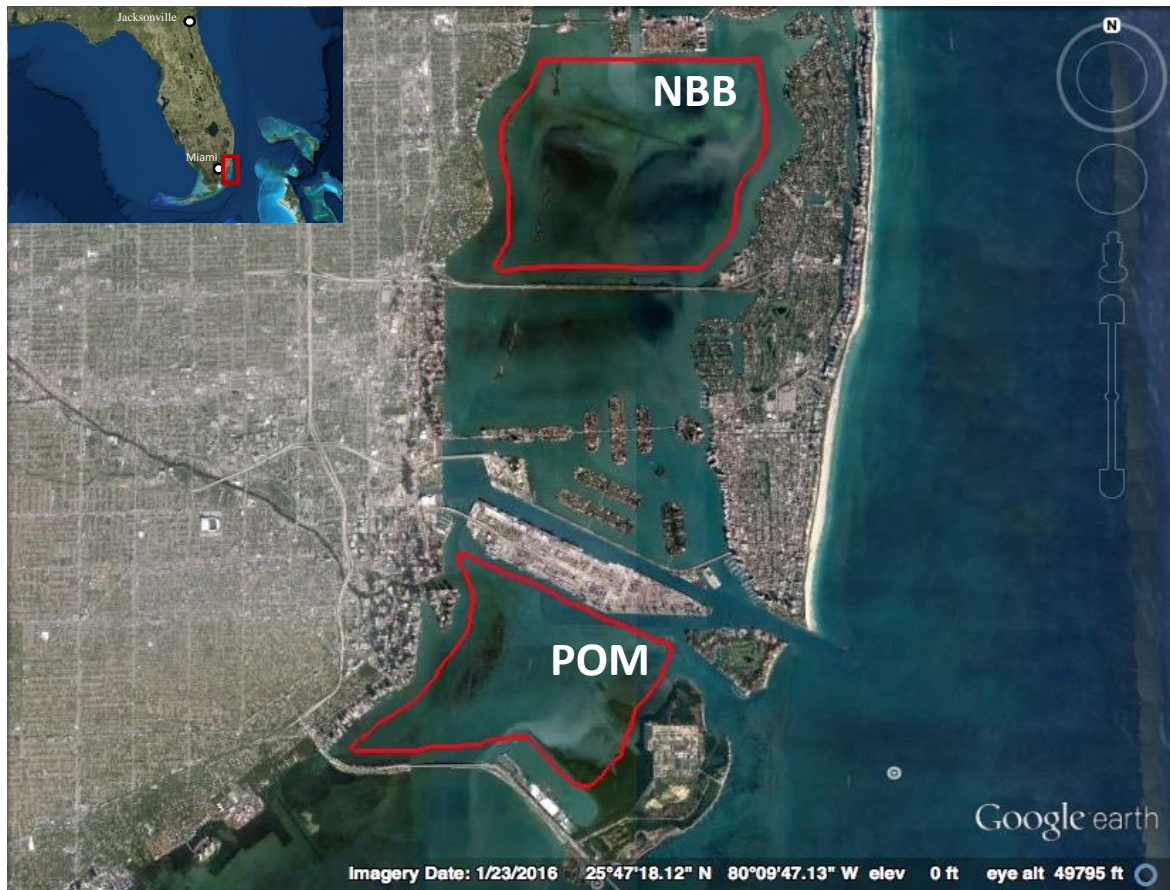
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## INTRODUCTION

The United States Geological Survey (USGS) and The National Oceanographic Atmospheric Administration (NOAA) were funded to monitor faunal changes in South Florida marine ecosystems. Data from two monitoring projects (Florida Invertebrate Assessment Network [FIAN, Robblee et al., 2014] and Faunal Monitoring in Response to Harbor Dredging [FMHD, Daniels et al., 2018]) were used to assess the extent to which seagrass cover correlates with fish community composition and structure at sites adjacent to the Port of Miami. These long-term data sets provided an opportunity to investigate possible impacts of variation in seagrass cover on seagrass-associated communities over time.

### **Background**

The FIAN project, a component of the Monitoring and Assessment Plan (MAP), was created to monitor 19 South Florida seagrass communities and associated fauna from 2005-2011 (Robblee and Browder, 2012; Robblee et al., 2014). Three years later, the FMHD project was implemented, which replicated the FIAN methods, and monitored faunal abundances in coordination with the harbor dredging of the Port of Miami (Daniels et al., 2018). Seagrass habitats and their associated fauna were sampled biannually before, during, and after dredging activity at two sites: North Biscayne Bay (NBB) and Port of Miami (POM) (Figure 1) (Daniels et al., 2018). Both FIAN and FMHD focused on collecting epibenthic fauna and surveying for submerged aquatic vegetation (SAV) in shallow intertidal and coastal ecosystems using a throw trap collection method and Braun-Blanquet method, respectively (Braun-Blanquet, 1931; Kushlan, 1981; Durako, 1988; Robblee and Browder, 2012; Daniels et al., 2018).



**Figure 1.** Two study sites outlined in red. Northern region is North Biscayne Bay (NBB) and the southern region is Port of Miami (POM). Site reference map of South Florida in top left corner. ("Port of Miami," 2016).

### Study Site

Both sites, NBB and POM are in the northern portion of Biscayne Bay Aquatic Preserve and have been used for recreational and commercial practices for many years (Johnson, 2011) (Figure 1). The Port of Miami, established in 1897, is a major international shipping hub and is characterized by heavy boat traffic and dense coastal development (Chapman, 1993; Robles et al., 2005).

### Physical Environment

Biscayne Bay Aquatic Preserve & Outstanding Florida Waters (OFW) is a Marine Protected Area, defined as “any area of the marine environment that has been reserved by federal, state, tribal, territorial, or local laws or regulations to provide lasting protection for part or all of the natural and cultural resources therein” (NOAA, 2009). This area is

managed by the Florida Department of Environmental Protection and NOAA's Coral Reef Conservation Program for the protection of wildlife inhabiting these marine areas (NOAA, 2009). Seagrasses, a type of SAV, account for 71% of the total benthic coverage within Biscayne Bay Aquatic Preserve and OFW and provide important habitats for estuarine fishes (Rozas and Odum, 1988; NOAA, 2009). These vegetated areas have a greater biodiversity than adjacent unvegetated environments (Jordan, 2002). Although monitoring programs such as FMHD have tended to focus on benthic juvenile fishes, shrimp, and decapod crustacean communities, the bay is home to many other taxa, such as large mammals and other macroinvertebrates (Daniels et al., 2018). This remarkable biodiversity has led to the designation of Biscayne Bay as a Habitat Focus Area (HFA) by NOAA; this designation is reserved for natural areas of high biological interest and value that are threatened by human activities (NOAA, 2016).

Like all coastal habitats, Biscayne Bay is subject to strong interactions with adjacent terrestrial ecosystems, and is therefore affected by human activities such as coastal urbanization and canalization. The effects on Biscayne Bay hydrology have been complex: for example, these activities have enhanced salinity fluctuations normally associated with the wet and dry seasons (Lohmann et al., 2012). Early 20<sup>th</sup> Century canalization generally decreased freshwater flow into the Bay, resulting in increases in salinity; but the canals and impermeable surfaces in urban areas have increased the input velocity and volume of storm runoff during the rainy season, leading to considerable variation in salinity (Browder et al., 2005; Quinones-Aponte, 2013). Overall there has been a general trend of increased salinity in NBB (USGS, 2004). Other water quality parameters have been affected as well, such as turbidity (Robles et al., 2005). Further urban development, climate change, and activities such as dredging (see below) are likely to lead to further unpredictability in this complex system, with potentially significant impacts on seagrass and associated fish communities in this region (Caccia and Boyer, 2007).

### ***Dredging of the Port of Miami***

The Port of Miami is a world-class cruise and trade port designed to accommodate the largest shipping vessels (USACE, 2004; Johnson, 2011). Initially, regular dredging and maintenance projects were conducted to maintain access to the Port (USACE, 2004). However, modifications to the Panama Canal produced an influx of larger and deeper

Panamax vessels, which required implementation of the Port of Miami Deep Dredge (PMDD) project to ensure their safe access to the Port (USACE, 2004; Barnes et al., 2015). The project proposed to dredge  $3.8\text{--}4.6 \times 10^6 \text{ m}^3$  of sediment from Miami Harbor, resulting in a deeper (by ~2-3m) and wider (up to 100 m) channel (Barnes et al., 2015). The project began in November 2013, with dredging of the inner portion of the Port during late 2014 and 2015 (USACE, 2011; USACE, 2013; Miller et al., 2016). Dredging of channels leading into NBB and POM altered salt and freshwater flows in and out of the bay, resulting in complex variations in water temperature, salinity, and turbidity (Robles et al., 2005).

### **Seagrass Communities and Ecological Significance**

According to the South Atlantic Fishery Management Council and National Marine Fisheries Service, seagrass beds constitute critical fish habitat (SAFMC, 1998; NOAA, 2000). Seagrass coverage has been shown to be positively correlated with greater fish and invertebrate abundance (Heck and Thoman, 1984; Orth et al., 1984; Rozas and Odum, 1988; Hughes et al., 2002). Seagrass beds provide protection and foraging areas, as well as nursery grounds for juvenile fishes (Sogard et al., 1987; Thayer et al., 1999). Seagrasses are beneficial to the economy as they provide habitat for important commercial fishes including various species of grouper, snapper, and mackerel (Idyll, 1999; Hill, 2002). Previous research has highlighted the importance of seagrass beds by demonstrating a decrease in water quality and abundance of commercial fishery stocks following a documented die-off (Robblee et al., 1991; Zieman et al., 1999; Lirman and Cropper, 2003). Monitoring and management of estuarine areas and water quality can increase our understanding in the overall value of seagrasses as well as acting as an indicator species providing insight to the health of coastal ecosystems (Hill, 2002).

### ***Species of Interest-Seagrass***

There are seven species of seagrasses found along the east coast of Florida including: *Thalassia testudinum*, *Syringodium filiforme*, *Halodule wrightii*, *Halophila decipiens*, *Halophila engelmanni*, *Ruppia maritima* and *Halophila johnsonii* (Green and Short, 2003). Preliminary analyses indicated that *T. testudinum* (turtle grass), *S. filiforme* (manatee grass) and *H. wrightii* (shoal grass) predominated in the study area, so all subsequent analyses involving SAV focused exclusively on those three species. *Thalassia testudinum*, the most abundance seagrass in the study area creates expansive beds of thick,

flat leaves on sandy and muddy, shallow water substrates (Fourqurean and Robblee, 1999; Green and Short, 2003). *Syringodium filiforme* is common among *Thalassia* beds, but can also be found as distinct, cylindrically shaped patches (Green and Short, 2003). Lastly, *H. wrightii* typically has a shorter canopy height than the previous species discussed and creates carpet-like beds in shallow waters (Green and Short, 2003). *Halodule wrightii* is a resilient species that can be found in the shallower waters close to shore and is able to tolerate salinities as high as 60-100 psu (Green and Short, 2003; Lirman and Cropper, 2003). When all three species are present, *T. testudinum* dominates and inhibits growth of the other two; however, varying environmental conditions, such as increased turbidity and nutrient input, can allow *S. filiforme* and *H. wrightii* to coexist with or even outcompete *T. testudinum* (Fong et al., 1997; Lirman and Cropper, 2003). Regardless of what taxa are dominant, SAV are widely distributed throughout the shallow, warm waters of Biscayne Bay, and provide critical habitat for many fishes (Short and Wyllie-Echeverria, 1996).

#### ***Environmental Parameters and Constraints: Seagrass***

Seagrasses require sunlight, suitable substratum, and wave energy (Greve and Binzer, 2004). Light drives the process of photosynthesis, resulting in growth of seagrass and high productivity (Durako, 1988). Seagrasses are found at various depths ranging from sea level to 90 m, but depth at which seagrasses can thrive is largely limited by light (Duarte, 1991). Suitable substratum is also necessary for the growth of seagrass (Greve and Binzer, 2004). Soft substrates, like mud and sand, allow for elongation of seagrass rhizomes and roots (Greve and Binzer, 2004). Lastly, seagrasses require moderate wave energy (Koch, 2001). Overly strong currents (>1.5 m per second) and wave activity inhibit seagrass growth because of increased sedimentation and turbidity (Greve and Binzer, 2004). Seagrass zonation typically follows suitable salinity gradients, water clarity, and substrate availability (Browder et al., 2005). Stress in seagrass ecosystems can be heightened by eutrophication and high turbidity from various activities in the surrounding area (Durako, 1988). Stress in the form of dredging, runoff from rivers and urbanized areas, and high boat traffic have the capability of altering salinity and light exposure to seagrasses, resulting in community health and abundance modifications (Durako, 1988).

Seagrass cover has declined due to natural and anthropogenic factors, specifically decreasing water clarity associated with high turbidity and nutrient loading (Short, 2005;

Erftemeijer and Robin Lewis, 2006). Changes in turbidity have also been associated with seagrass die-offs and shifts in the dominant seagrass taxa (Zieman et al., 1999). In one study, seagrass beds decreased in biomass by 28% in a 10-year time span (1984-1994) (Forqurean and Robblee, 1999). Florida Bay has experienced an extended history of seagrass loss and environmental impacts, including a massive seagrass die-off in 2015 that was characterized by increased salinity and water temperatures (National Park Service, 2016). This can lead to reduced density of seagrass-associated organisms that use seagrass beds for protection and feeding (McCloskey and Unsworth, 2015).

### **Fish Communities**

Fish species found throughout Biscayne Bay include grunts (Haemulidae), killifish (Fundulidae/Cyprinodontidae), seahorses and pipefish (Syngnathidae), and gobies (Gobiidae). Previous work in the study area suggest that the most abundant fish species inhabiting the study area were canopy dwellers that use SAV for shelter from predation, or as substrate for foraging (Powell et al., 1987; Matheson, Jr. et al., 1999). *Thalassia testudinum* has been shown to offer cover for smaller juvenile fishes while providing foraging grounds for larger predators (Heck and Orth, 1980). Some seagrass-associated fishes live their entire life in this habitat; consequently, they are dependent on its overall health and productivity (Matheson, Jr. et al., 1999). Diversity and abundance of juvenile and adult fishes have been correlated with SAV density (Heck and Orth, 1980). The link between the fishes and their preferred SAV substrate is so well-established that variations in fish community composition and structure can be used to infer the health of their SAV substrate, and of the larger marine areas they inhabit (Heck and Orth, 1980; Powell et al., 1987).

### ***Species of Interest- Fish***

Preliminary analyses in the study area (see below) indicated that SAV fish community structure was largely driven by five key fish species: *Lucania parva* (Rainwater killifish), *Gobiosoma robustum* (Code goby), *Microgobius gulosus* (Clown goby), *Floridichthys carpio* (Goldspotted killifish) and *Hippocampus zosterae* (Dwarf seahorse). They are all characterized as benthic fish species and are considered benthic carnivores that feed on larval crustaceans, amphipods, and benthic invertebrates including polychaete worms (Masterson, 2008; Sweat, 2009). Although they have similar habitat preferences, *L.*

*parva* and *H. zosterae* are defined as resident species within a seagrass community, while *G. robustum*, *M. gulosus*, and *F. carpio* are transient species that have the capability to change their ecosystem preference at some point in their life (Matheson, Jr. et al., 1999). Since 2007, these five species were found every year throughout the study area, indicating communities of these fishes reside in NBB and POM most of the time.

### ***Environmental Parameters and Constraints: Fish***

Studies on the physiological preferences of these fishes suggest that their abundance is correlated to the abiotic conditions in their habitat, such as salinity, water temperature, or turbidity. For example, although *Lucania parva* are the most abundant fish in Biscayne Bay, they are largely restricted to marine waters; conversely, *M. gulosus* is occasionally found in more brackish habitats due to its tolerance of wider ranges of salinity values (Foster, 1967; Schofield, 2003; Barimo and Serafy, 2003). *Gobiosoma robustum* is also found in high abundances throughout Biscayne Bay but is almost exclusively found where conditions are favorable for SAV to flourish (Schofield, 2003). As with *L. parva*, *F. carpio* has been found in relatively high abundances within Biscayne Bay and Florida Bay regions but is much more resilient to temperature fluctuations than the others listed (Kaill, 1967; Gilmore et al., 1978; Sogard et al., 1987). It can tolerate temperatures ranging from 6°C to 37.7°C (Kaill, 1967; Gilmore et al., 1978). Some studies have found *F. carpio* in highest abundance during the height of the dry season in South Florida, with densities exceeding 3 individuals per m<sup>2</sup> (Sogard et al., 1987). *Hippocampus zosterae* depend heavily on near-shore habitats dominated by seagrass and macroalgae that are typically characterized by minimal wave energy and currents, which is critical for the survival and success of this species (Fedrizzi et al., 2015). *Hippocampus zosterae* use the vegetation as a substrate for anchoring, as well as a protected environment for settling in at birth and during the time they are too small to latch onto the vegetation (Fedrizzi et al., 2015).

## **STUDY RATIONALE, SCOPE, AND HYPOTHESES**

Using long-term data sets (2007-2011; 2014-2016), this project examined fluctuations in fish community composition and structure in relation to variation in seagrass cover and selected critical environmental variables (temperature, salinity, and turbidity). This study had the following specific objectives:



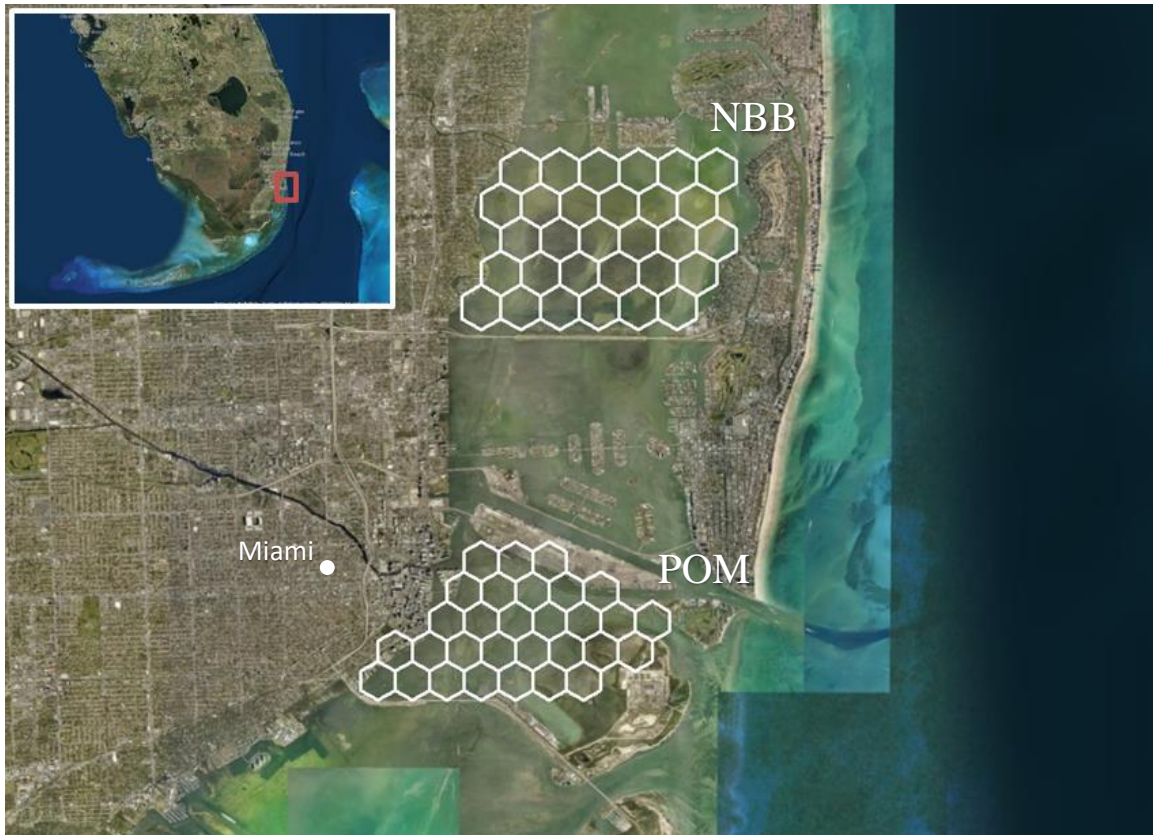
1. Quantitatively assess how fish community similarity varied among years and among sample sites, using a multivariate approach.
2. Assess how SAV community composition, coverage, and environmental variables varied among sites and years.
3. Explore the extent to which variation in fish community was correlated with SAV or environmental variables.

Biological communities are inherently dynamic, so the null hypothesis of “no community change” is unlikely to be falsified under natural conditions and therefore, biologically uninteresting (Peters, 1991). Thus, all analyses were conducted to establish whether fish community structure a) varied significantly among sites and years but more importantly, b) might be statistically correlated to SAV community composition and coverage, and environmental variables.

## MATERIALS AND METHODS

Data sets from 2007-2011 (FIAN) and 2014-2016 (FMHD) were used for analysis. These data sets are publicly available online at: <https://doi.org/10.3133/ofr20181052>. (April 12, 2018). Spring and fall samples were collected each year from two regions (NBB and POM) during April and September. The materials and methods remained consistent throughout the eight years of data collection and were adopted from FIAN, a previous study conducted in South Florida (Robblee and Browder, 2012; Daniels et al., 2018).

The original sampling plan for FIAN consisted of 19 sample sites throughout South Florida (two of which were used for this study; NBB & POM), which were subdivided into 30 equal-sized hexagonal cells (Robblee and Browder, 2012) (Figure 2). Within each cell, six random locations were sampled for the presence of seagrass and/or benthic algae. If SAV was present, it was identified to species and using the Braun-Blanquet Method, a semi-quantitative (on a scale of 0 to 5; see table 1) estimate of percent coverage was recorded (Braun-Blanquet, 1932; Robblee et al., 2014). The six samples in each hexagonal cell were then averaged, yielding mean coverage values for each seagrass taxon within that cell.



**Figure 2.** Sampling grid cells situated in NBB and POM (30 cells per site). Sample area outlined in red in the South Florida reference map (ESRI, 2012).

Cover Score	Description
0	No vegetation present
0.1	Solitary shoot
0.5	Sparse coverage
1	0-5% Coverage
2	5-25% coverage
3	25-50% Coverage
4	50-75% Coverage
5	75-100% Coverage

**Table 1.** Scale used from the Braun-Blanquet method for assigning % cover of SAV present at sample site. Cover scores range from 0-5 and are paired with % cover ranges from 0-100%.

Divers noted the sediment texture of the area (sand, mud, muddy sand, sandy mud, *Halimeda* hash, coarse shell, rubble, or a combination) and the species of vegetation present in each quadrat. Canopy height (cm) of seagrasses present was measured, unless algae were the only species present, then algal height was measured. Seagrass and algae were identified using Green and Short (2003) and Eiseman, N. J (1980). Data collected in FIAN between 2005 and 2006 were not used for analysis in this project due to modifications in Braun-Blanquet methods being expanded from three to six replicates in the year 2007.

## Fish Collection

A 1 m<sup>2</sup> throw-trap was used at each of the sampling locations. The trap was thrown from the stern of the boat into undisturbed water (Figure 3). When it settled on the bottom, the weighted nylon netting attached at parallel sides of the trap was stretched over the top to capture the organisms within.



**Figure 3.** A 1 m<sup>2</sup> throw-trap being deployed from the stern of the boat. The trap was used for collection of benthic fishes with sweep nets (Photo by Candace Grimes).

After the trap and netting was in place, fishes were collected using sweep nets; as these nets were dragged along the bottom inside the trap, the nylon netting attached to the trap was stretched behind the sweep nets to limit escapes. Once a net was pulled through, it was removed and taken to the boat. Each trap was swept five times. In sample locations with a depth greater than 0.75 m, SCUBA (surface-supplied hookah) was used. The five sweep nets were rinsed on site and samples passed through a 1-mm sieve (Figure 4). Samples were rinsed into separate small mesh bags and stored in an ice bath. Samples were then brought to the lab and fixed in a 10% Formalin solution for at least 72 hours before



sorting. Fishes were then sorted by sample location and stored in 70% Ethanol for identification. Fishes were identified to lowest taxon possible using a dissecting microscope while referencing Atlantic Coast Fishes Field Guide (Robins and Ray, 1999). Fishes damaged or too small were identified to family and listed as “unknown” species. The samples were then placed in labeled bags for storage in 70% Ethanol.



**Figure 4.** *The sweep nets being rinsed on board and ran through a 1-mm sieve to collect all organisms before placing them in bags for storage (Photo by Elizabeth Colhoun).*

### **Environmental Measurements and Techniques**

Sample location was determined using a Global Positioning System (GPS) that was predetermined in the lab by randomly selecting a single point within each of the 60 cells. At each location seven environmental measurements were taken including: salinity and water temperature from top and bottom of water column, turbidity, sediment depth, and water depth. For this study only water temperature, turbidity, and salinity values were used for analysis. Salinity and water temperature at the surface and bottom of the water column were taken using a hand-held WTW 330i Conductivity Field Meter. A water sample was

collected from undisturbed water just below the surface to test for turbidity. The sample was stored on ice upon return to the lab and an HF Scientific DRT-15CE portable turbidity meter measuring nephelometric turbidity units (NTU) was used. Collections in 2015 and 2016 used an YSI meter (6600 V2-4 Multi-Parameter Sonde) to capture salinity, temperature and turbidity in the field directly. For analysis, the average of the surface and bottom measurements were used for salinity and temperature comparisons. Sediment depth was measured at the throw-trap sample point by probing a 3 m long metal rod marked in 1 cm increments until the hard substrate was reached. Water depth was measured using a 3 m PVC pole also marked in 1-cm increments. All measurements and methods can be found in Table 2.

Parameter	# of Replicates Per Cell	Range	Method
Sediment Texture	6	N/A	Estimate: S=sand, M=mud, MS=muddy sand, SM=sandy mud, HH= <i>Halimeda</i> hash, CS=course shell, R=rubble, or a combination
Canopy Height	6	0-193 cm	Maximum average height (cm)
Cover/Abundance Estimate	6	N/A	% cover of seagrass and algae by species; 0.25m <sup>2</sup> quadrat.
Surface/Bottom Salinity	1	17.40-42.15 PSU	Hand-held WTW 315i and 330i Conductivity Meter; YSI (6600 V2-4 Multi-Parameter Sonde) (2015 & 2016 ONLY) (PSU)
Surface/Bottom Temperature	1	19.0-34.0 °C	Hand-held WTW 315i and 330i Conductivity Meter; YSI (6600 V2-4 Multi-Parameter Sonde) (2015 & 2016 ONLY) (Celsius)
Water Turbidity	1	0.19-15.20 NTU	DRT 15C Turbidimeter; YSI (6600 V2-4 Multi-Parameter Sonde) (2015 & 2016 ONLY) (NTU)
Sediment Depth	1	0-360 cm	3-meter probe, sediment surface to bedrock (cm)
Water Depth	1	23.2-884.0 cm	3-meter PVC pole; water surface to sediment surface at time of sampling (cm)

**Table 2.** *Habitat and environmental measurements and methods associated with FIAN and FMHD field collections. Parameters include: sediment texture, canopy height, cover/abundance estimate, salinity, water temperature, turbidity, sediment depth, and water depth.*

## Structure of Dataset

To summarize, the overall structure of the data set included data on the fish community (abundance of individual taxa), SAV community (percent cover of individual taxa), and three environmental predictor variables, averaged from six sampling locations within each of 30 cells at two distinct locations.

## Statistical Methods

Multivariate analysis was conducted in PRIMER-E v.7.0 (Clarke et al., 2014; Clarke and Gorley, 2015); PRIMER routines are indicated in bold. Basic QA/QC was conducted using the **CHECK** function to identify data entry errors and missing data. Fish community data included information on 86 fish species that were caught and identified from NBB and POM, the majority of which were only occasionally sampled or constituted a minor part of the overall fish community. The data set was therefore simplified using Distance-based Linear Modeling (**DistLM**), which identified a core group of 25 fish taxa that most contributed to variations in community structure in the data set (Table 3). These 25 fish taxa tended to be the most abundant from all samples over eight years. However, some of these taxa had not been identified to species (or even to family) and were therefore excluded from further analysis. Furthermore, preliminary analysis also indicated that the abundance of many of the taxa were strongly correlated to one another; therefore a second **DistLM** procedure was conducted to further reduce the data set, resulting in the subset of five fish species that accounted for 80% of the overall variation in community structure, as listed above (see Table 4). All subsequent fish community analyses were conducted on this representative subset.

Fish abundance was square root transformed, and used to calculate Bray-Curtis similarity indices for all pairs of samples. The Bray-Curtis index estimates community composition similarity among sites: it ranges from zero to one, with zero indicating no species in common, and one indicating that the two communities being compared have exactly the same species with the same abundance. A dummy variable in the Bray-Curtis was used to allow inclusion of samples where no fish were found. The result was a triangular matrix of similarities that was the basis for all subsequent tests. The **PERMANOVA** procedure was first used to test for fish community differences due to location, year, and season.



A SAV similarity matrix was established based on three seagrass species (*H. wrightii*, *T. testudinum*, and *S. filiforme*) using the same approach as for the fish community. Environmental predictor variables (water temperature, salinity, turbidity) were all measured at different scales. Therefore, these data were first standardized, and then normalized so that all values ranged from -1 to 1 and averaged zero. An environmental variable similarity matrix was then established for all pairs of samples using Euclidean distance.

The **RELATE** procedure (a modified Mantel test) was then used to examine correlations of the environmental predictor variables with fish community structure. **RELATE** returns a test statistic, rho ( $\rho$ ), that ranges from -1 to +1; one indicates full correlation between the similarity patterns in the fish community dataset and the predictor variables, whereas zero indicates no correlation. **RELATE** also provides a p value for this test statistic. A second **RELATE** test was used to explore correlations with SAV. A **BIOENV** test with a post-hoc forward stepwise regression was used to identify the best models (if any); model information content was estimated using AIC (Akaike Information Criterion) statistic (AICc).

### ***Ordination***

Fish community data were ordinated using Nonmetric Multidimensional Scaling (**nMDS**), to graphically represent fish community similarity patterns in the data. The initial **nMDS** included 720 data points and was difficult to interpret; to improve clarity and facilitate interpretation; the ordination was simplified using the **Distance Between Centroids** procedure, which depicted the centroids for the data (grouped by location, year, and season). A second **nMDS** was established using these centroids. The resulting graph included two overlays: Pearson correlation vectors for seagrass and/or environmental variables, and community similarity clusters determined using the **CLUSTER** procedure (group average). Similar ordinations were established for the environmental data and the SAV community.

## RESULTS

Distance based linear modeling identified a subset of five fishes that accounted for 80% of the variability in the complete fish community data set: these were Fundulidae *Lucania parva*, Gobiidae *Microgobius gulosus*, Gobiidae *Gobiosoma robustum*, Cyprinodontidae *Floridichthys carpio*, and Syngnathidae *Hippocampus zosterae* (Table 4).

DistLM				
Variable	R <sup>2</sup>	Pseudo-F	P	Cumul.
Fundulidae <i>Lucania parva</i>	0.39909	18.569	0.001	0.39909
Gobiidae unknown unknown	0.46892	3.5503	0.001	0.46892
Gobiidae <i>Microgobius gulosus</i>	0.53739	3.8477	0.002	0.53739
Gobiidae <i>Gobiosoma robustum</i>	0.59119	3.2903	0.002	0.59119
Cyprinodontidae <i>Floridichthys carpio</i>	0.63913	3.1883	0.001	0.63913
Syngnathidae <i>Syngnathus scovelli</i>	0.67724	2.7154	0.012	0.67724
Haemulidae <i>Haemulon sp</i>	0.70955	2.448	0.018	0.70955
Gobiidae <i>Coryphopterus glaucofraenum</i>	0.73955	2.4183	0.015	0.73955
Gobiidae <i>Ctenogobius stigmaturus</i>	0.77004	2.6516	0.01	0.77004
Syngnathidae <i>Hippocampus zosterae</i>	0.7992	2.7593	0.014	0.7992
Haemulidae <i>Haemulon aurolineatum</i>	0.8253	2.69	0.011	0.8253
Callionymidae <i>Diplogrammus pauciradiatus</i>	0.85116	2.9537	0.014	0.85116
Gobiesocidae <i>Gobiesox punctulatus</i>	0.8727	2.7061	0.033	0.8727
Syngnathidae <i>Syngnathus sp</i>	0.89308	2.8601	0.034	0.89308
Cynoglossidae <i>Symphurus pelicanus</i>	0.91117	2.8502	0.033	0.91117
Gobiidae <i>Ctenogobius boleosoma</i>	0.92487	2.3716	0.09	0.92487
Unknown Unknown fish	0.93773	2.4767	0.055	0.93773
Lutjanidae <i>Lutjanus cyanopterus</i>	0.94888	2.4008	0.069	0.94888
Batrachoididae <i>Opsanus beta</i>	0.95954	2.6329	0.045	0.95954
Lutjanidae unknown unknown	0.96921	2.8271	0.076	0.96921
Atherinidae <i>Atherinomorous stipes</i>	0.9771	2.7563	0.047	0.9771
Syngnathidae <i>Anarchopterus criniger</i>	0.98359	2.7712	0.066	0.98359
Gobiidae <i>Microgobius sp</i>	0.98898	2.9307	0.071	0.98898
Syngnathidae <i>Cosmocampus albirostris</i>	0.99345	3.4147	0.08	0.99345
Labrisomidae <i>Paraclinus fasciatus</i>	0.99779	7.8675	0.041	0.99779
Labrisomidae <i>Paraclinus marmoratus</i>	1	-12.892	0.968	1.0007

**Table 3.** Distance-based Linear Modeling in PRIMER-E of top 25 fish taxa from all samples. These most contributed to variations within the community structure and tended to be most abundant during the eight years of sampling.

DistLM				
Variable	R <sup>2</sup>	Pseudo-F	P	Cumul.
Fundulidae <i>Lucania parva</i>	0.39909	18.596	0.001	0.39909
Gobiidae <i>Microgobius gulosus</i>	0.53739	3.8477	0.001	0.53739
Gobiidae <i>Gobiosoma robustum</i>	0.59119	3.2903	0.002	0.59119
Cyprinodontidae <i>Floridichthys carpio</i>	0.63913	3.1883	0.001	0.63913
Syngnathidae <i>Hippocampus zosterae</i>	0.7992	2.7593	0.014	0.7992

**Table 4.** Distance-based Linear Modeling in PRIMER-E showing a subset of 25 fishes from Table 3. The five species (*Fundulidae Lucania parva*, *Gobiidae Microgobius gulosus*, *Gobiidae Gobiosoma robustum*, *Cyprinodontidae Floridichthys carpio*, and *Syngnathidae Hippocampus zosterae*) account for 80% of the variability within the fish community data.

The **PERMANOVA** model detected significant effects of (in decreasing order of effect size) location (Pseudo-F =112.1, p=0.001), year (Pseudo-F = 14.2, p=0.001), and season (Pseudo-F = 3.0, p=0.037). There was also a significant interaction between location and year (Pseudo-F = 7.5, p=0.001), indicating that although community structure varied between locations and among years, the response to year differed between locations. Season did not have a significant interaction with location or year (p=0.137; p=0.144, respectively) (Table 5). Thus, POM and NBB fish communities changed differently during the course of the study (Table 5).

PERMANOVA Table of Results			
Source	df	Pseudo-F	P(perm)
Season	1	2.9702	0.037
Location	1	112.1	0.001
Year	7	14.173	0.001
Season x Location	1	1.9423	0.137
Season x Year	7	1.338	0.144
Location x Year	7	7.5148	0.001
Season x Location x Year	7	1.087	0.35
Res	928		
Total	959		

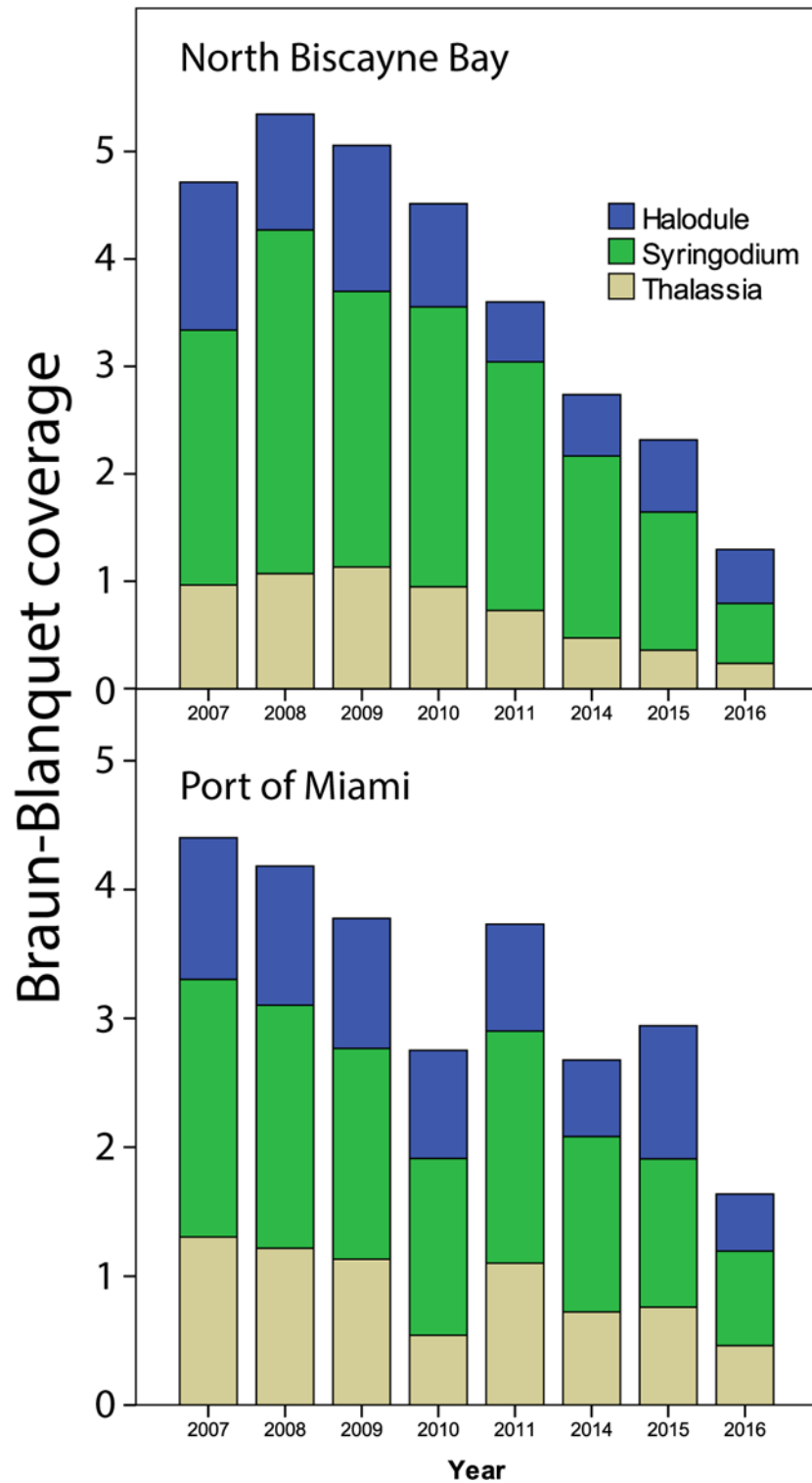
**Table 5.** A PERMANOVA test from PRIMER-E displaying the significance of the top five fish taxa between season, location, and year.

A **PERMANOVA** (Table 6) was done on seagrass cover abundance between year and location and resulted in a significant correlation (Pseudo-F=2.5377) (Figure 5). This

trend can also be seen in the stacked bar graph in Figure 5 that plots individual seagrass taxa abundance against year. A decreasing trend can be seen, suggesting a loss of seagrass for both locations and years. **BIOENV** and **BVSTEP** tests suggest *Syringodium* was the individual taxa responsible for 65% of the variation within the data ( $\rho > 0.95$ ).

<b>PERMANOVA Table of Results</b>			
<b>Source</b>	<b>df</b>	<b>Pseudo-F</b>	<b>P(perm)</b>
Location	1	16.458	0.001
Year	7	13.985	0.001
Location x Year	7	2.5377	0.001
Res	944		
Total	959		

**Table 6.** A PERMANOVA test from PRIMER-E displaying the significance of seagrass cover abundance between location and year.



**Figure 5.** Stacked bar graph from PRIMER-E representing the average seagrass cover by species (*Syringodium*, *Halodule*, and *Thalassia*) between years.

To compare fish communities, seagrass communities, and environmental parameters to one another the **RELATE** function was used on original resemblance matrices from transformed fish abundance data. **RELATE** finds the correlation of ranked similarities within matrices and produces a sample statistic or Rho. Rho can be any number ranging from -1 to +1 and is most significant when the sample statistic is close to a value of 1. The **RELATE** function revealed a significant correlation between fish abundance and seagrass cover abundance (Rho=0.326). Fish abundance compared to environmental parameters produced a Rho value of -0.002 suggesting there was no significance (Table 7). When **RELATE** was used to compare environmental data and seagrass cover data a significant correlation was found (Rho=0.355) (Table 7).

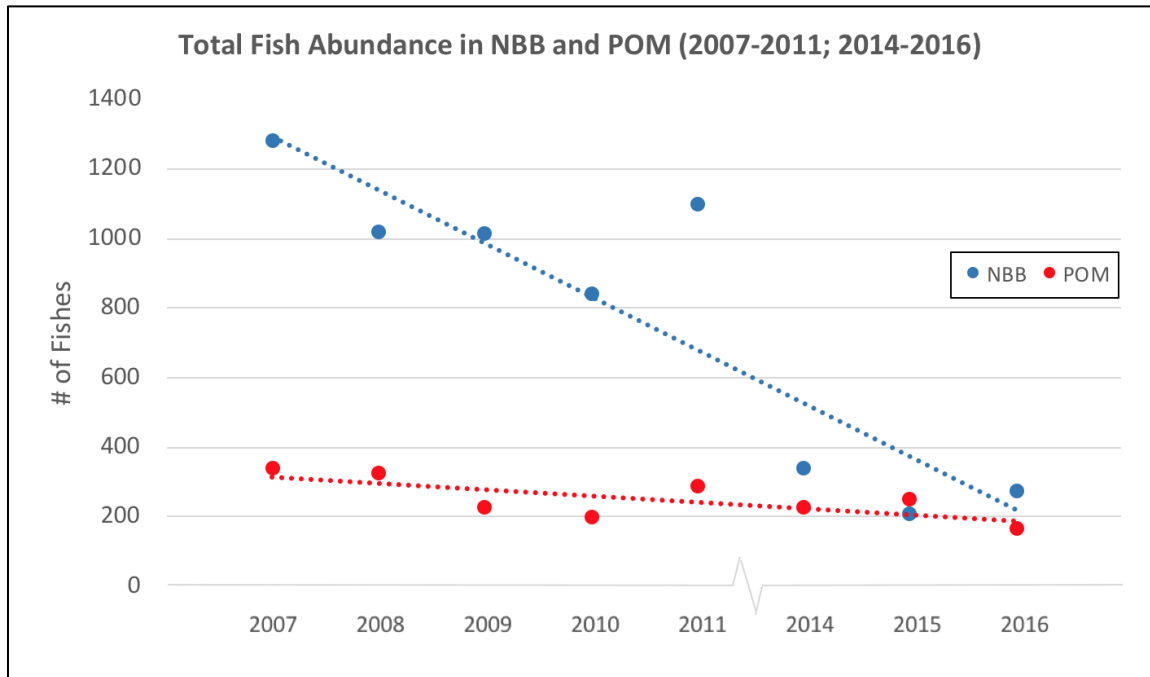
<b>RELATE</b>		
<b>Variable</b>	<b>Sample Statistic (Rho)</b>	<b>Sig. level of sample stat.</b>
Fish -> Seagrass	0.326	1.20%
Fish -> Environmental	-0.002	45.40%
Environmental -> Seagrass	0.355	0.80%

***Table 7.** RELATE function in PRIMER-E was used to find the correlation between fish communities, seagrass communities, and environmental parameters.*

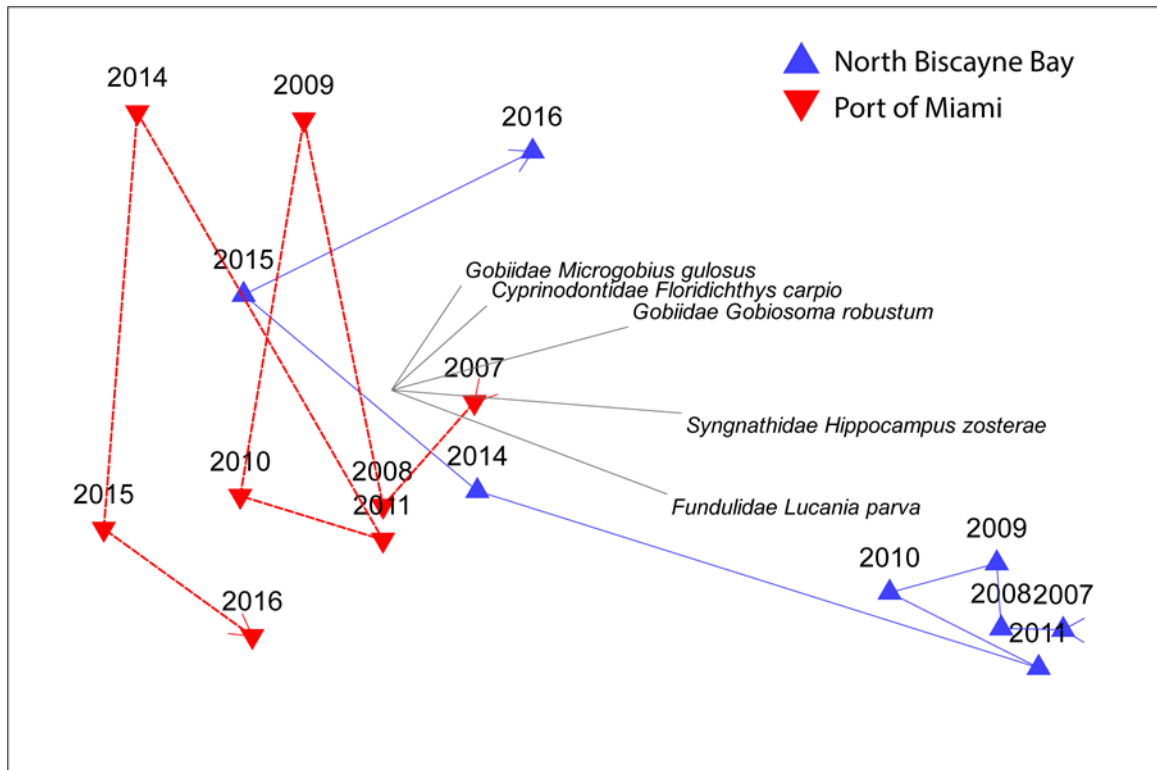
## **Ordination**

Abundance statistics revealed an overall decreasing trend in the abundance of all fish species collected in both NBB and POM throughout the study (Figure 6). Ordination of the fish communities at both locations over time indicated that NBB and POM had distinct fish communities, but that this distinction decreased over time, so that by 2014 the distinctness of the NBB fish community had been negated (Figure 7). Pearson correlation vectors for each taxon indicated that, as with SAV, this transition over time was marked by the loss of species (Figure 7 & 8). Community similarity for three SAV taxa (*Syringodium*, *Halodule*, and *Thalassia*) at each location over time is depicted in Figure 8. Patterns suggesting highest abundance are represented by Pearson correlation vectors that point in the direction of the most abundance taxa. Ordination indicated a clear shift in SAV community structure throughout the duration of the study, marked by a steady decrease in SAV coverage at both locations, particularly NBB (Figure 8). Ordination of environmental

parameters indicated that the sites varied in their water temperature, salinity, and turbidity, although no clustering or obvious trends were evident (Figure 9).

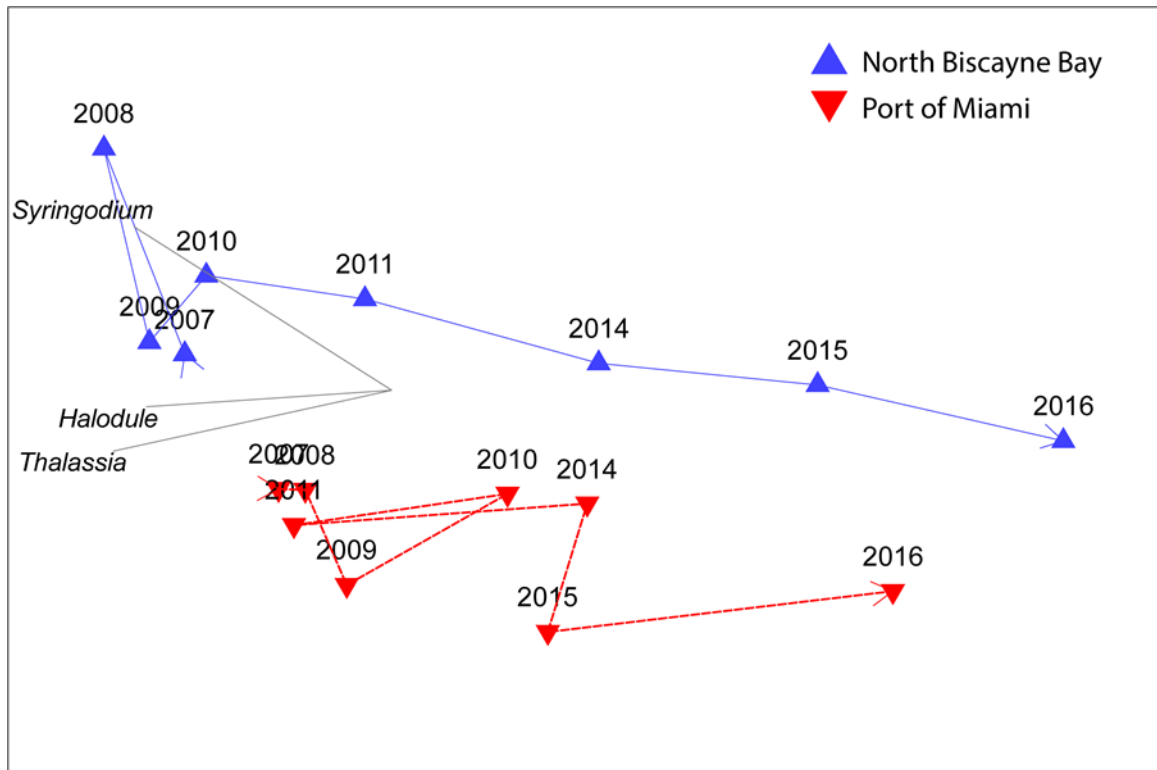


**Figure 6.** Fish abundance statistics represented by a scatter plot separated by location, with negative trend lines. This graph shows the overall abundance of all fish species collected between 2007-2011; 2014-2016.

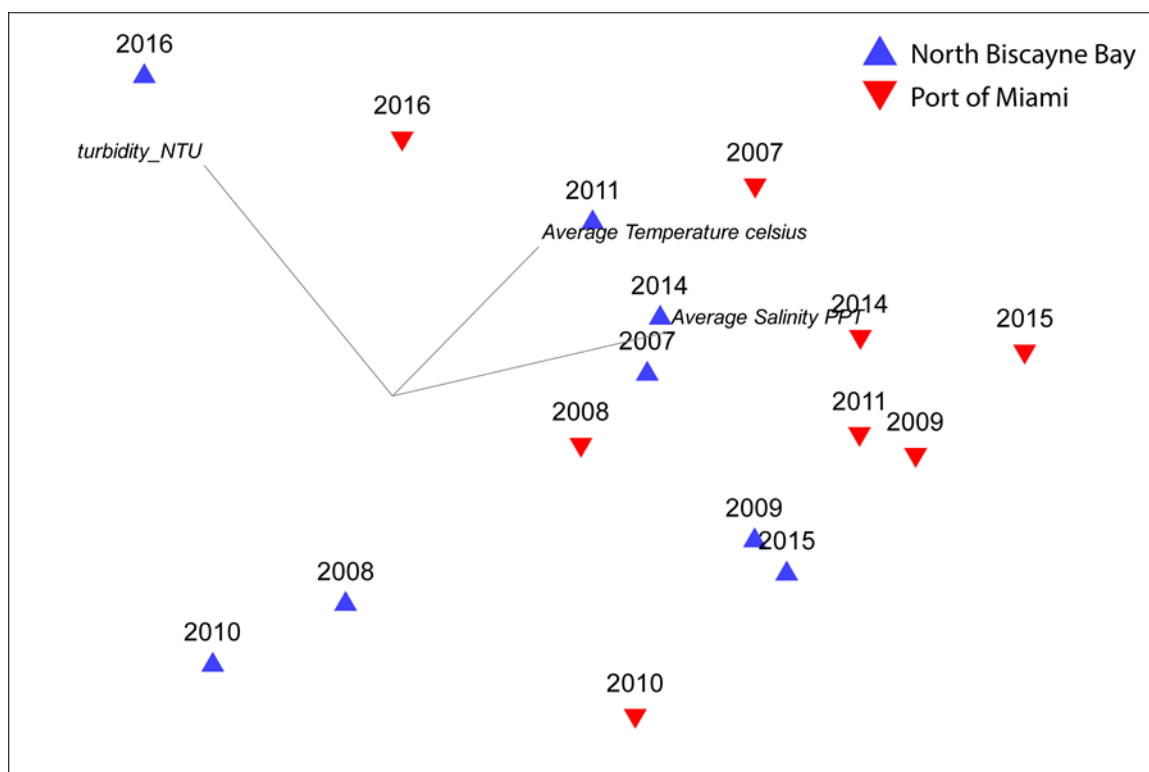


**Figure 7.** Ordination from PRIMER-E, displaying the five fish species (*Fundulidae Lucania parva*, *Gobiidae Microgobius gulosus*, *Gobiidae Gobiosoma robustum*, *Cyprinodontidae Floridichthys carpio*, and *Syngnathidae Hippocampus zosterae*), at both locations (NBB & POM), over time (2007-2011; 2014-2016). Fish species are represented as Pearson correlation vectors with a 2D stress value = 0.03.





**Figure 8.** Ordination from PRIMER-E, displaying the three seagrass species (*Thalassia*, *Halodule*, and *Syringodium*) abundances at both locations (NBB & POM), over time (2007-2011; 2014-2016). The seagrass species are represented as Pearson correlation vectors with a 2D stress value = 0.05.



**Figure 9.** Ordination from PRIMER-E, displaying the three environmental parameters (turbidity, salinity, and water temperature), at both locations (NBB & POM), over time (2007-2011; 2014-2016). The environmental parameters are represented as Pearson correlation vectors with a 2D stress value = 0.12.

## DISCUSSION

The key conclusion from the above data is that there were major changes in fish community structure over time that were largely driven by loss of species and increased homogenization of the fish communities at both locations. More specifically, the NBB community shifts to resemble that of POM by 2014. These changes in fish community mirrored significant changes in SAV, again largely driven by the loss of vegetation cover at both locations.

Data from FIAN and FMHD (USGS) provided an opportunity to examine changes to seagrass cover and associated fish communities at two locations in South Florida (NBB and POM). This study compared fish abundance in NBB and POM to seagrass cover and various environmental variables. Although there was a time gap between the conclusion of FIAN (2007-2011) and the beginning of FMHD (2014-2016), gradual changes in fish and

SAV community structure were observed. Five fish species (*L. parva*, *M. gulosus*, *G. robustum*, *H. zosterae* and *F. carpio*) and three of the dominant subtropical/tropical seagrass species in the Western Atlantic (*H. wrightii*, *T. testudinum*, and *S. filiforme*) were chosen for comparative purposes, and their abundance varied significantly throughout the duration of the study.

Seagrasses are adapted to live in environments with high light exposure and low-nutrient concentrations (Fong et al., 1997). North Biscayne Bay (NBB) and POM are dominated by *T. testudinum* but successfully co-exist with *H. wrightii*, *S. filiforme* and other algae and macroalgae, such as *Halimeda*, *Caulerpa*, and *Gracilaria* (Fong et al., 1997; Green and Short, 2003). Changes in environmental conditions affect the three species differently. Studies have shown *H. wrightii* is the best adapted to wide ranges of salinity, even thriving at salinities up to 72 psu, while *S. filiforme* is the most restricted of the three (Lirman and Cropper, 2003; Green and Short, 2003). *Thalassia testudinum* alternatively, will stop growing when exposed to extreme salinity values, such as 45-60 psu (Lirman and Cropper, 2003). Salinity values during this study ranged from 17.4-39.7 psu in NBB and 20.77-42.15 psu in POM. While none of the recorded salinity values fell within these extremes, without more frequent salinity measurements, we cannot definitively say they never reach these high values (Montague and Ley, 1993). Water temperature and turbidity can limit flora and fauna survival in an ecosystem, but in this study both temperature and turbidity remained relatively constant throughout sampling, and likely had little, if any impact (Greve and Binzer, 2004). The results indicate that SAV cover is being affected by environmental factors other than those studied here. While it is important to understand the dynamics of environmental variables and their effects on fauna, we must be cautious when interpreting the data in this study. Although a long-term data set has been compiled for these environmental conditions, measurements at a specific location and only twice a year, may not provide an accurate estimate of the ecosystem as a whole.

Although the exact causes of reduction in seagrass cover and environmental changes in NBB and POM are unknown, dredging activities and hydrology have direct effects on environmental conditions and survival of seagrass-associated species (Fong et al., 1997; Lirman and Cropper, 2003). Statements directly from the US Army Corps Management Plan report possible impacts to seagrasses in the areas surrounding dredging

activity to be “unavoidable”, suggesting impacts are inevitable, but the extent of those impacts remain poorly studied (USACE, 2004). Although dredging impacts may be unavoidable, prior to the PMDD project (2007-2014), a decline in fish abundance was already being documented. Due to the timing, this could be linked to reasons other than dredging. From a study done in a Western Australia sound, in the years between 1954-1978 seagrass meadows decreased from 4200 to 900 ha and occurred during an increase of industrial development on the shore surrounding the sound (Cambridge and McComb, 1984). North Biscayne Bay (NBB) and POM are in direct contact with the city of Miami and the associated industrial infrastructure, which could be associated with the gradual decrease in fish abundance. Further studies are required to assess the extent to which ongoing dredging activities and other factors might be affecting SAV cover, thereby indirectly affecting fish communities as well.

The shift seen in abundance statistics of individual fishes by location and year indicates a distinct change took place during the years of 2011-2014 (Figure 7). These years coincide with several events including: the Port Miami Deep Dredge (PMDD) project during the years 2013-2015; periods of drought; and major storm events (Barnes, 2015). Tropical Storm Isaac (2012) and Andrea (2013) produced abnormally high rainfall amounts for the area, resulting in flooding and increased runoff, as well as higher than normal amounts of salt water pulsing into the bay from storm surge (NWS, 2017). Droughts and documented seagrass die-off, can lead to a change in environmental conditions that can persist for several years (National Park Service, 2016). Increased amounts of storm water runoff can change sedimentation, turbidity, and salinity, which can cause species composition and trophic position of the fish community to shift towards a less seagrass-dependent community (Erftemeijer and Lewis, 2006). This can also change the dominant fish species in the community (Fourqurean and Robblee, 1999; Thayer et al., 1999). *Lucania parva* was by far the most abundant species in this study within NBB and POM and similar community compositions have been recorded in other regions (Matheson, Jr. et al., 1999). Many studies investigating overall ecosystem health have been implemented using fish community structure, due to their correlated responses to contaminants and implications as indicator species (Loeb and Spacie, 1994). Overall, trends of *L. parva* are continuing to decline in similar seagrass dominated ecosystems like the North Biscayne

Aquatic Region for reasons including: loss of habitat for foraging and protection (Sogard et al., 1987; Lubbers et al., 1990; Matheson, Jr. et al., 1999). Loss of *L. parva* and its high abundance within NBB and POM could result in new species being introduced and *L. parva* populations decreasing. It also could indicate natural processes, like storms and rainfall could be impacting the habitat this species resides in.

North Biscayne Bay and POM's environmental conditions are extremely variable and biological species are being forced to evolve in order to survive. The explanation to changes in seagrass and fish community structure between the years 2011-2014 is not known, but a multitude of factors come into play. Regions with more frequent events including: maintenance dredging, boating, and runoff can alter the sediment composition and result in a new or altered make-up of the biological community composition within that ecosystem (Newell et al., 1998). An adjustment in the sediment composition can create a slower recovery time for seagrass communities that come in contact with disturbances. This has been seen in some marine benthic communities that have muddy/sandy sediment texture, and results in slower recovery times in comparison to communities with clean sandy areas (Dernie et al., 2003). A study looking at recovery times of seagrasses and nekton took dredged material and placed it in a seagrass community area to see how these factors would respond and how long it would take them to recover, if at all (Sheridan, 2004). They found the water column and sediment factors settle out at a rate of 1.5 to 3 years, whereas seagrasses and nekton were predicted to take 4 to 8 years in order to recover (Sheridan, 2004). This study suggests the rate at which maintenance dredging takes and the material the seagrass has available to recolonize on, can impact the recovery success of seagrasses and nekton living within the seagrass community (Sheridan, 2004).

Disturbances, depending on the frequency and duration, can have a positive or negative effect on an ecosystem (Short and Wyllie-Echeverria, 1996). Highest biodiversity in some cases has been explain using the intermediate disturbance hypothesis which suggests, small-scale disturbances can in fact have a positive effect on an ecosystem, by increasing species diversity and providing new areas for species recolonization (Connell, 1978; Thistle, 1981). Recovery time and the scale of the disturbance can limit the species that are able to recolonize or survive in the area (Thistle, 1981; Dernie et al., 2003). The species in question and its physical condition must be taken into account as well (Short and

Wyllie-Echeverria, 1996). For example, in many reviews, seagrass leaves have been recorded breaking off or whole plants being up-rooted by wind driven disturbances, but these are instances where recovery is achievable relatively quickly (Short and Wyllie-Echeverria, 1996). On the other hand, wind-induced turbidity can have long lasting effects resulting in possible seagrass die-off due to decreased light penetration (Robblee et al., 1991; Short and Wyllie-Echeverria, 1996). In some cases, after an initial negative impact, an ecosystem may slowly recover, however only if the initial stressor has been removed (Bilkovic, 2010). If a disturbance takes place at a rate more frequent than the time it takes for a habitat to recover, the long-term success of the habitat will not be assured (Sheridan, 2004). Due to the various disturbances NBB and POM face, seagrass and associated fish communities could be impacted and result in lower biodiversity because there is not adequate time for recovery.

Multiple reports of seagrass loss in the US have been documented including: a loss of 25,220 ha of seagrass in Tampa Bay, FL in response to mechanical dredging, and discharge from sewage and industrial construction, and a loss of 14,000 ha in Laguna Madre, TX from increased turbidity in response to frequent maintenance dredging (Lewis et al., 1985; Quammen and Onuf, 1993; Short and Wyllie-Echeverria, 1996). These are just a few of the examples of how impactful consistent disturbances can be on seagrass communities. Seagrass growth is dependent upon the conditions to which it is exposed to, which makes it difficult to recover from loss unless conditions improve or change to favor a more resilient seagrass species (Durako, 1988; Greve and Binzer, 2004). It is unknown if the fish communities in Biscayne Bay will rebound or revert to a new fish community, but the direction of recovery will be dependent upon many biotic and abiotic factors (Dernie et al., 2003). Continual and frequent maintenance dredging activities in the bay, in addition to increased growth and development around the water's edge will continue to put stress on the bay. Over time, increased impermeable surfaces leading to runoff, anthropogenic influences such as dredge activity and boating, and intense storm episodes have altered both NBB and POM and the area will continue to evolve.

### **Future studies**

Due to funding, this project was not continued past collections in fall of 2016. This restricted the possibility of detecting any rebound in either seagrass cover or fish

abundance. Additional monitoring of community structure and seagrass cover could reveal how this ecosystem responds to anthropogenic activities like dredging and natural events including major storms and changes in rainfall. If sampling continued we would be able to make more hypotheses as to what is impacting fish and seagrass community structure within NBB and POM. Investigating other factors including runoff or water management spillways, water depth, and sediment depth could provide us a better understanding of what is happening in this region. Funding for additional monitoring to mirror the data collection of FIAN and FMHD has been requested. Gaining additional funding is crucial due to recent reports from NOAA stating they fear the urban bay may be on track to reaching its “tipping point,” or point of no return, in reference to its water quality (NOAA, 2016). In 2016, Biscayne Bay became part of a focus area and the implementation plan went into review, but how it will precede is still in the works (NOAA, 2016). This dedication of Biscayne Bay could be a step in protecting and preserving critical fish communities and SAV. Understanding how Biscayne Bay and the surrounding areas change and evolve could lead to improvements in dredging efforts, coastal development, and management plans in order to protect benthic fish communities and their habitat within coastal ecosystems.

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